

Inclusion of soil erosion impacts in life cycle assessment on a global scale: application to energy crops in Spain

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Received: 26 February 2012 / Accepted: 25 October 2012 / Published online: 16 November 2012
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Abstract

Purpose Despite the fundamental role of ecosystem goods and services in sustaining human activities, there is no harmonized and internationally agreed method for including them in life cycle assessment (LCA). The main goal of this study was to develop a globally applicable and spatially resolved method for assessing land use impacts on the erosion regulation ecosystem service.

Methods Soil erosion depends much on location. Thus, unlike conventional LCA, the endpoint method was regionalized at

the grid cell level (5 arcmin, approximately $10 \times 10 \text{ km}^2$) to reflect the spatial conditions of the site. Spatially explicit characterization factors were not further aggregated at broader spatial scales.

Results and discussion Life cycle inventory data of topsoil and topsoil organic carbon (SOC) losses were interpreted at the endpoint level in terms of the ultimate damage to soil resources and ecosystem quality. Human health damages were excluded from the assessment. The method was tested on a case study of five 3-year agricultural rotations, two of them with energy crops, grown in several locations in Spain. A large variation in soil and SOC losses was recorded in the inventory step, depending on climatic and edaphic conditions. The importance of using a spatially explicit model and characterization factors is shown in the case study.

Conclusions The regionalized assessment takes into account the differences in soil erosion-related environmental impacts caused by the great variability of soils. Taking this regionalized framework as the starting point, further research should focus on testing the applicability of the method through the complete life cycle of a product and on determining an appropriate spatial scale at which to aggregate characterization factors in order to deal with data gaps on the location of processes, especially in the background system. Additional research should also focus on improving the reliability of the method by quantifying and, insofar as it is possible, reducing uncertainty.

Responsible editor: Llorenç Milà i Canals

Electronic supplementary material The online version of this article (doi:10.1007/s11367-012-0525-5) contains supplementary material, which is available to authorized users.

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Keywords Ecosystem services · Land use impacts ·
Regionalized life cycle impact assessment · Soil organic
carbon · Soil loss

1 Introduction

Life cycle assessment (LCA) aims to provide a general picture of the environmental impacts of resource consumption and

emissions during the entire cycle of a product system. LCA normally uses spatially and temporally independent linear impact assessment (LCIA) models to calculate the environmental damage caused by the resource consumption and emissions quantified in the life cycle inventory (LCI) of the product system under analysis. Site-specific conditions are therefore not usually taken into account when the environmental impacts of the product are evaluated. The impacts of the different LCA categories have consequences for the environment and human welfare on different spatial scales. The spatial scale at which these consequences occur has nothing to do with the importance of the categories, but with a need for spatial differentiation for some impact categories. Since economic processes are spread worldwide, local impacts have a global extension as well (UNEP 2003).

Despite the fundamental role of ecosystem goods and services in sustaining human activities, such as climate regulation, nutrient cycling, and erosion control (Millennium Ecosystem Assessment 2005), there is no harmonized and internationally agreed method for including them in LCA. According to a key framework for land use impact assessment in LCA (Milà i Canals et al. 2007a), ecosystem goods and services that should be integrated within LCA are impacts on biodiversity and, at least, impacts on the following five major ecosystem services: biotic production potential, carbon sequestration potential, freshwater regulation potential, water purification potential, and erosion regulation potential. Operational characterization factors and methods have been developed for impacts on biodiversity (De Schryver et al. 2010; Koellner and Scholz 2008) and on ecosystem services (Brandão and Milà i Canals 2012; Müller-Wenk and Brandão 2010; Saad et al. 2011). The methodology for assessing the degradation of the erosion regulation function due to land transformation and land occupation was developed using different spatial scales (Saad et al. 2011). However, erosion regulation characterization factors (CFs) were developed within a Canadian context and therefore are not globally applicable (Saad et al. 2011).

Because an activity's land use impacts depend on local conditions, a conventional site-independent LCA methodology might not be accurate. Methods focusing on these impacts should therefore include geospatial information in both the LCI and LCIA phases. Different levels of regionalization (e.g., countries, ecoregions, and biomes) and ecological unit classifications (e.g., life zones by Holdridge 1947 and ecoregions by Olson et al. 2001) in the LCIA of a range of land use impacts are presently used without a clear recommendation on a standardized approach to address spatial differentiation. Although the use of ecological or geographical units instead of administrative borders provides better estimates of the site dependency of land use impacts, especially in countries with a high degree of

variability, it is generally easier to find information to create regionalized impact assessment CFs at the country scale.

Our objective was to go one step further toward the integration of ecosystem services in LCA by developing a globally applicable and spatially resolved method to include land occupation impacts on the erosion regulation. Indicators of the impact category were defined on the endpoint level and were modeled up to the entities described by the areas of protection (AoP), i.e., natural resources, natural environment, and human health. The case study conducted to demonstrate the applicability of the method focused on the impacts of agricultural rotations with energy crops in Spain as compared to the cultivation of rotations with only traditional crops.

2 Materials and methods

Figure 1 illustrates the general land use impact mechanism and shows the three impact pathways studied as follows:

Land occupation leads to soil erosion, and this leads to loss of topsoil reserves, which leads to soil resource depletion (AoP natural resources, impact pathway 1 in Fig. 1).

Land occupation leads to soil erosion and altered soil function, which affects net primary production and leads to damage to ecosystem quality (AoP natural environment, impact pathway 2 in Fig. 1).

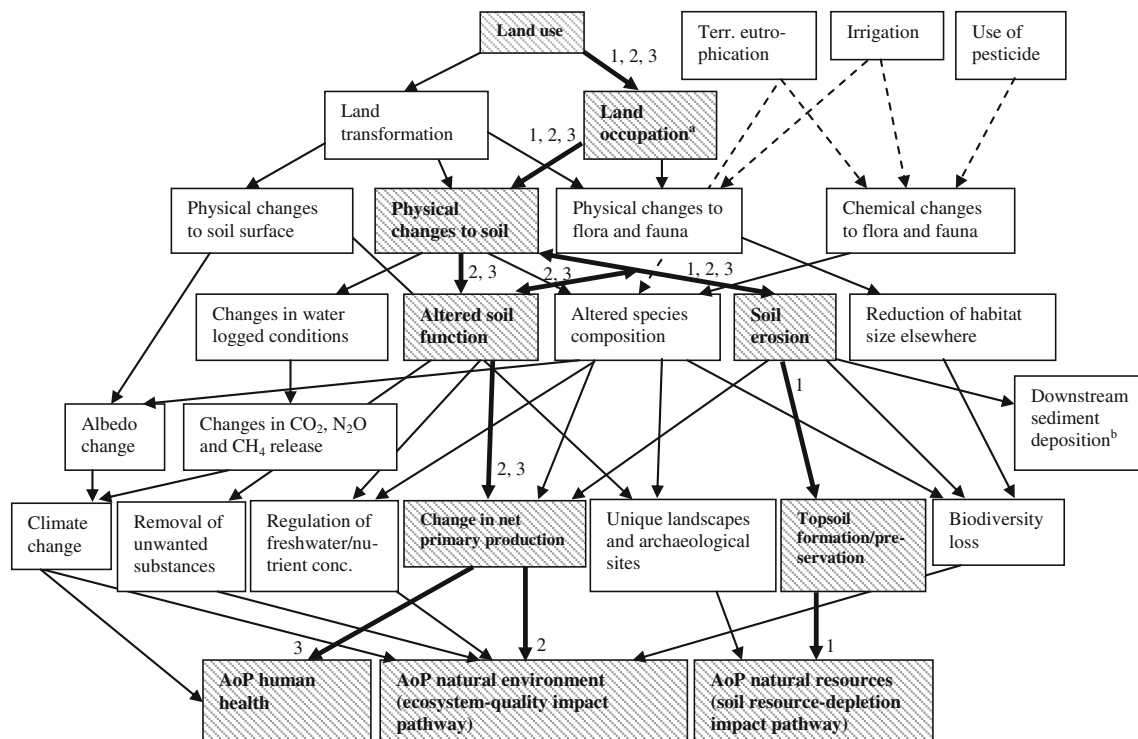
Land occupation leads to soil erosion and altered soil function, which affects net primary production and leads to damage to human health (AoP human health, impact pathway 3 in Fig. 1).

2.1 Soil resource depletion impact pathway

This AoP represents damage from the impact of removal of soil resources from the environment on soil resource depletion (impact pathway 1 in Fig. 1). Annually, humans cause the loss of 50–75 billion metric tons of soil (Harvey and Pimentel 1996). Agricultural land accounts for 75 % of the soil erosion worldwide, though it also occurs in other human-modified ecosystems, such as during the construction of roads and buildings. More than 75 % of the arable soils of the world suffer from moderate to very high soil losses (Reich et al. 2001), generally ranging from 10 to 100 $\text{tha}^{-1}\text{year}^{-1}$ (Pimentel et al. 1987), which is at least tenfold higher than the average rate of soil formation of 0.5–1 $\text{tha}^{-1}\text{year}^{-1}$ (Mann et al. 2002). Current soil losses due to land use (land occupation) reduce soil availability as a future resource.

2.1.1 Life cycle inventory data requirements

The type of land use has a determining role in the quantity of soil loss as specific direct physical interventions are often



^a Land occupation does not “cause changes” but contributes to prolong altered conditions

^b Benefits from downstream deposition of eroded material disregarded in the paper, as sediments cross the system boundary

Fig 1 Main impact pathways related to land use. The pathways discussed in the paper are shown with *cross-hatching and thicker arrows* (adapted from the ILCD handbook; JRC 2010)

related to land management (Frischknecht et al. 2007). In LCA, the ecoinvent database v3.0 (Weidema et al. 2011), based on Global Land Cover 2000 (Bartholomé and Belward 2005) and ecoinvent v2.0 (Frischknecht et al. 2007), identifies ten main types of land use and land cover classes in the first level (e.g., agriculture, forest) and provides more detailed information on land intensity and management in the following levels (e.g., arable non-irrigated and arable non-irrigated extensive). This tiered structure allows for different levels of detail in the LCI, depending on the quality requirements of the LCA study and the information available on the product under study. We chose this flexible classification system to record the land occupation type in the LCI. Yet, ecoinvent v3.0 does not distinguish between cultivated crops and specific crop management practices (e.g., type and timing of tillage operations and use of residue mulches), which are key factors controlling soil erosion on agricultural lands.

Soil losses due to land occupation must also be included in the LCI as soil loss mass (in grams). There are many estimation models that can be used to estimate soil loss, such as PESERA (Gobin and Govers 2003), INRA (Le

Bissonais et al. 2002), or universal soil loss equation erosion model (USLE; Wischmeier and Smith 1978). From all of them, we recommended the USLE to register soil losses in the LCI. USLE takes into account the effect of a particular land use type on soil erosion by water. There is a consensus that the USLE equation and its update (revised USLE, RUSLE; Renard et al. 1997) are valid methods to estimate soil losses by water at the inventory stage (Beck et al. 2010; Muys and García Quijano 2002). The use of (R)USLE to predict soil erosion losses has increased worldwide due to the growing availability of and accessibility to climatic, edaphologic, and land use and land cover data at the local and regional levels. This spatial information can be increasingly found in a geo-referenced format, thus allowing for data processing and visualization in a geographic information system (GIS) software. Furthermore, there are many biophysical models, such as EPIC (Williams et al. 1984) or APEX (Williams and Izaurralde 2005), which already have algorithms to simulate soil erosion with (R)USLE for many crops and at different spatial resolutions. Results from scenario simulations from these models can be incorporated in the inventory stage to register soil erosion.

Soil formation was not contrasted to soil erosion in this study because factors other than human land use, namely, climate, and the soil parent material, are generally recognized as the most important aspects governing soil formation (Jenny 1994). Data on the area as well as the duration required for the production of a certain amount of products and services have to be collected in the LCI as land occupation impacts are recorded in area per time (e.g., in square meters per year). The geo-referenced location of land use (longitude/latitude) should also be included if available. Failing that, a broader resolution (e.g., region, country) can be used, although this reduces the quality of the LCI data by increasing uncertainty due to the high variability of spatial conditions.

In summary, the following elementary flows need to be accounted in LCI: type of land use, soil erosion by water, time of occupation, area of occupation, and location of the activity.

2.1.2 Impact assessment model

Similar to other endpoint methods (Ecoindicator 99, EI99; Goedkoop and Spriensma 2001), damage to resources is expressed as surplus energy needed to make the resource available at some point in the future. This is a suitable unit to evaluate soil depletion, which indicates the anticipated energy removal from nature to provide a unit of soil eroded during land occupation. Instead of using energy units (megajoule equivalents), such as in EI99, we used emergy units (megajoule solar equivalents). Emergy is the environmental work needed for the formation of a natural resource (Odum 1996). In emergy, solar energy is the unit of reference as it is the primary energy source on the earth. Unlike the energy metric, emergy accounts for quality differences of the energy used to generate a product or service by converting raw units (e.g., kilograms soil and cubic meter water) to a common basis, i.e., units of solar energy. For example, sunlight, soil, water, and coal are assessed together by expressing them all with the solar energy unit. The advantage of emergy compared with other energy and exergy units is that it not only accounts for energy carriers (e.g., gas) and non-energetic materials (e.g., minerals) but also assigns an exergy (useful energy) value to land use (Rugani et al. 2011). Therefore, emergy evaluates the (solar) energy the natural system is deprived of to yield the new stock of soil lost during land use. We assigned an average transformity (i.e., the emergy required to make available one unit of energy, mass, or volume of the resource) for all soil types and locations: $23.9 \text{ MJ}_{\text{se}}\text{g}^{-1}$ soil (Odum 1996). This means that around 24 MJ of solar energy is required to generate a gram of soil lost by erosion. In LCA, transformity has been called solar energy factor (SEF; Rugani et al. 2011). The effect of soil erosion on soil resource depletion (ΔR) is expressed as follows with units of megajoule solar

equivalent soil loss per unit of area and time of land occupation:

$$\begin{aligned} \text{If soil loss} = 0, \quad \Delta R &= 0 \\ \text{If soil loss} > 0, \quad \Delta R &= \underbrace{A \times t \times \text{Soil loss}}_{\text{LCI}} \times \underbrace{\frac{SD_{\text{ref}} - SD_i}{SD_{\text{ref}}} \times \text{SEF}_{\text{soil}}}_{\text{CF}} \\ &= \text{MJ}_{\text{se}}\text{m}^2\text{year} \end{aligned} \quad (1)$$

This endpoint indicator combines the inventory flow (i.e., soil loss) with the local available soil reserves (SD_i , soil depth in the specific location i) on a spatial resolution of 5 arcmin (approximately $10 \times 10 \text{ km}^2$; FAO/UNESCO 2007) and with the solar energy factor of soil (SEF_{soil}) as the characterization factor. Soil loss mass of the LCI is weighted with soil depth as the environmental significance of soil loss depends on the soil stock size at the specific location. Twenty-one soil depth classes were distinguished based on the FAO's soil depth classification map, from very shallow soils (0.05 m deep) to very deep soils (2.25 m deep). These classes resulted from the combination of five major soil depth categories (further details in Electronic supplementary material (ESM), Section 1). Local soil reserves on each grid cell of the soil depth map were normalized with a reference soil depth (SD_{ref}). We selected the upper limit of the very deep soil depth category as the reference ($SD_{\text{ref}}=3 \text{ m}$) as an indicator of the potential soil quality. This would only occur if the whole of the grid cell area had this maximum soil depth, which we assumed was not possible once the first soil particle was eroded. Note, therefore, that SD_i can never be equal to SD_{ref} , so the characterization factor always takes positive values and any amount of soil loss will have an environmental impact. Choosing the maximum soil depth as the reference was judged to be a more representative correction of the site dependency of the characterization factor rather than the soil reserves of a particular region (e.g., Swiss lowlands for ecosystem quality assessment in the EI99 methodology; Goedkoop and Spriensma 2001). Larger damage factors were assigned to thinner soils by calculating the difference between SD_{ref} and SD_i before normalization with the soil reference. This means that, based on the same amount of soil loss, thinner soils are more vulnerable than thicker soils. The indicator relates impact assessment to biogeographical conditions in each grid cell i (SD_i) without any further aggregation of land use type or land use cover. For a land use activity under study, the lower the indicator result, the less soil resource depletion and the less damage to the environment.

2.2 Ecosystem quality impact pathway

This AoP represents damage from the negative impacts of altered soil function and net primary productivity on the

function and structure of natural ecosystems (impact pathway 2 in Fig. 1). In the erosion process, soil quality declines, essential plant nutrients are lost, and soil depth is reduced. As a result, biomass productivity diminishes. Ultimately, this can adversely affect overall biodiversity and ecosystem quality. Numerous positive correlations between plant biomass productivity (NPP) and vascular plant species diversity and richness have been established (Costanza et al. 2007; Flombaum and Sala 2008), though overall biodiversity does not always correlate with productivity (e.g., Mediterranean hot spots and intensively managed agricultural lands).

One of the methods used in LCA for measuring environmental impacts on ecosystem functions is based on the soil organic matter content (Milà i Canals et al. 2007b). We assessed the effects of soil erosion on the terrestrial ecosystem quality by, in a first step (damage factor), linking soil loss to soil organic carbon (SOC) loss and, in a second step (effect factor), linking SOC loss to biomass production drop. Thus, ecosystem biomass production was modeled as a function of soil quality, which is indicated by the SOC content of the soil lost. We assessed ecosystem biomass production as a function of the net primary production of potential natural vegetation (NPP₀, i.e., the anticipated state of mature vegetation in the absence of human intervention).

2.2.1 Life cycle inventory data requirements

SOC losses must be registered in the LCI. Like soil erosion, SOC losses can be directly derived using biophysical models like EPIC (Williams et al. 1984) or APEX (Williams and Izaurrealde 2005). Alternatively, SOC losses may be calculated by, in a first step, determining the topsoil OC content and, in a second step, multiplying the already estimated soil losses by the percentage of topsoil OC content. The quantity of SOC in the topsoil can be determined by direct measurements. In the absence of plot-level soil data, SOC content can be determined using already existing spatial data layers (e.g., map of organic carbon in topsoils in Europe: Jones et al. 2005; Harmonized World Soil Database, HWSD: FAO et al. 2009), though inventory data quality will be affected. On the other hand, while site-specific measurements are more accurate, this is not likely to be achievable in most LCA studies (Milà i Canals et al. 2007b).

The soil unit where the impact activity occurs should be included in the LCI using the most recent FAO classification (FAO et al. 1990), which identifies 28 different soil units (soil types), each with harmonized soil parameters. The soil unit can be identified by direct measurements on the occupied land under analysis. Otherwise, it may be approximated using literature or spatial databases such as the Harmonized World Soil Database (FAO et al. 2009). Again, direct measurements are more accurate, although not always available in LCA studies.

To estimate the SOC in each soil unit, we determined the content of the 28 soil units in the FAO classification system (Table 1). We averaged the topsoil OC content of over 16,000 soil mapping units covering the entire land area of the world with a resolution of 30 arcsec (approximately $1 \times 1 \text{ km}^2$) in the HWSD (FAO et al. 2009).

The 28 soil units were grouped into seven major categories (see Table 1). In 24 out of 28 soil units, there was <2 % SOC, the threshold selected by the European Commission for defining soils in phase of pre-desertification (COM 2002). As in the case of the inventory for the soil resource depletion indicator, information on the type and intensity of land use, area size (in square meters), duration (in years), and the location of the occupation should be recorded.

2.2.2 Impact assessment model

According to the International Reference Life Cycle Data System (ILCD) Handbook (JRC 2010), which is a series of technical documents that provide detailed guidance on all the steps required to conduct a LCA study, species diversity is the recommended indicator to be implemented in end-point LCA methodologies when modeling damage to ecosystem quality. Effects on species diversity are usually quantified in terms of the potentially disappeared fraction (PDF) of vascular plant species (Koellner 2000), such as in the EI99 (Goedkoop and Spriensma 2001), ReCiPe 2008 (Goedkoop et al. 2009), and IMPACT2002+ (Jolliet et al. 2003) methods. However, according to ILCD, function-related parameters, such as the biomass production of the ecosystem, might also be good endpoint indicators.

A limited number of studies have focused on accounting for current or potential NPP losses caused by soil erosion due to the complex connection between the two factors. In addition to soil properties, NPP depends on many other physical environmental aspects, such as leaf area index, precipitation, atmospheric CO₂ concentration, and temperature (Melillo et al. 1993). NPP is therefore commonly used as an indicator to reflect ecosystem response to climate change. Change in climate may decrease NPP (lower precipitation or cloudiness) or increase it (photosynthesis enhancement). Due to the many relationships between factors, estimates of the direct cause–effect chain between NPP and soil erosion are scarce and highly uncertain.

Of the studies performed on the relationship between soil loss and NPP/NPP₀ loss, most estimate productivity losses according to qualitative degrees of erosion (e.g., light and extreme) and limit the scope to the local or the regional level (Mann et al. 2002; Mokma and Sietz 1992). We developed an initial approach to convert soil loss–NPP₀ loss into approximate quantitative linear relationships using data at

Table 1 Topsoil organic carbon (percent weight) of the 28 soil units in the Soil Map of the World (FAO et al. 1990), according to the HWSD (FAO et al. 2009), and linear equations used in the impact model

Soil units HWSD	Topsoil organic carbon (% weight)	%NPP ₀ depletion equations	
Gypsisols	0 to <0.5	If SOC _{loss} = 0,	NPPD = 0
Arenosols		If SOC _{loss} > 0,	NPPD = 4.09 × SOC _{loss} + 2.66 for SOC _{loss} < 23.80 g
Calcisols			NPPD = 100 for SOC _{loss} ≥ 23.80 g
Solonchaks	0.5 to <1.0	If SOC _{loss} = 0,	NPPD = 0
Lixisols		If SOC _{loss} > 0,	NPPD = 1.96 × SOC _{loss} + 2.66 for SOC _{loss} < 49.66 g
Luvisols			NPPD = 100 for SOC _{loss} ≥ 49.66 g
Solonetz			
Plinthosols			
Planosols			
Fluvisols			
Regosols	1.0 to <1.5	If SOC _{loss} = 0,	NPPD = 0
Leptosols		If SOC _{loss} > 0,	NPPD = 1.32 × SOC _{loss} + 2.66 for SOC _{loss} < 73.74 g
Acrisols			NPPD = 100 for SOC _{loss} ≥ 73.74 g
Vertisols			
Cambisols			
Anthrosols			
Kastanozems			
Ferralsols	1.5 to <2.0	If SOC _{loss} = 0,	NPPD = 0
Greyzems		If SOC _{loss} > 0,	NPPD = 0.88 × SOC _{loss} + 2.66 for SOC _{loss} < 110.61 g
Podzoluvisols			NPPD = 100 for SOC _{loss} ≥ 110.61 g
Alisols			
Nitisols			
Phaeozems			
Chernozems			
Gleysols	2.0 to <2.5	If SOC _{loss} = 0,	NPPD = 0
Podzols		If SOC _{loss} > 0,	NPPD = 0.69 × SOC _{loss} + 2.66 for SOC _{loss} < 141.07 g
			NPPD = 100 for SOC _{loss} ≥ 141.07 g
Andosols	4.86	If SOC _{loss} = 0,	NPPD = 0
		If SOC _{loss} > 0,	NPPD = 0.31 × SOC _{loss} + 2.66 for SOC _{loss} < 314.00 g
			NPPD = 100 for SOC _{loss} ≥ 314.00 g
Histosols	34.60	If SOC _{loss} = 0,	NPPD = 0
		If SOC _{loss} > 0,	NPPD = 0.04 × SOC _{loss} + 2.66 for SOC _{loss} < 2433.50 g
			NPPD = 100 for SOC _{loss} ≥ 2433.50 g

Soil units within the same soil category are arranged by increasing SOC content. SOC_{loss} in the %NPP₀ depletion (NPPD) equations is expressed as grams of C loss (in a square meter and year)

the global level found in the literature (Dregne and Chou 1992; FAO/UNEP 1984; Zika and Erb 2009). This was done for each soil unit group in Table 1 (see the procedure in the ESM, Section 2). Using this method, soil loss was first related to the loss of SOC as a measure of soil quality and, finally, to ecosystems' loss of biomass productivity.

In the impact assessment model, the inventory flow (i.e., SOC losses) should be used as an input parameter in the equation in Table 1 for the soil unit where the land use activity is developed. Note that for SOC losses greater than or equal to a predetermined threshold for each soil unit group (when NPPD=100), NPP₀ is completely lost and the soil is unlikely to be able to recover. Soils with

low SOC content, such as those in arid and semi-arid areas, are less resilient than soils rich in SOC, which are usually found in wet regions. These SOC loss thresholds are equivalent to a soil depletion of approximately 65 t ha⁻¹year⁻¹, which is extreme soil loss. Most agricultural land in the world loses soil at a rate of between 13 and 40 t ha⁻¹year⁻¹ (Pimentel and Kounang 1998), whereas losses of more than 100 t ha⁻¹year⁻¹ only occur in extreme events (Morgan 1992).

The effects of soil erosion on ecosystem quality (ΔEQ) are expressed using a growth-based value: NPPD (potential net primary production depletion). For an occupation of 1 m² and 1 year, ΔEQ ranges from 0 to 1 (percentage expressed as a decimal).

$$\begin{aligned}
 &\text{If } \text{SOC}_{\text{loss}} = 0, \quad \Delta \text{EQ} = 0 \\
 &\text{If } \text{SOC}_{\text{loss}} > 0, \quad \Delta \text{EQ} = \underbrace{A \times t \times \frac{a \text{SOC}_{\text{loss}} + b}{100}}_{\text{LCI}} \times \underbrace{\frac{\text{NPP}_{0,i}}{\text{NPP}_{0,\text{ref}}}}_{\text{CF}} \\
 &\quad = \text{NPPD m}^2\text{year}
 \end{aligned} \quad (2)$$

This endpoint model combines the inventory flow ($a \times \text{SOC}_{\text{loss}} + b$, if $\text{SOC}_{\text{loss}} > 0$, i.e., mass of SOC losses transformed into %NPP₀ losses) with NPP₀ values spatially resolved for each grid cell ($\text{NPP}_{0,i}$) at 5 arcmin (approximately $10 \times 10 \text{ km}^2$; Haberl et al. 2007) to obtain the absolute biomass productivity drop at the specific location. Values were then normalized with an NPP₀ value corresponding to that of the ecosystem with the highest biotic productivity worldwide ($\text{NPP}_{0,\text{ref}} = 1,496 \text{ gC m}^{-2} \text{ year}^{-1}$) as a representative reference of the potentiality of the ecosystem. Larger impact factors were allocated to the most productive soils (higher $\text{NPP}_{0,i}$). From an ecosystem services viewpoint, we made the general assumption that ecosystems with higher NPP values are considered more valuable as NPP is a scarce resource on earth (Pfister et al. 2011). The most productive lands can also be used for a greater diversity of purposes. Unlike Eq. 1, $\text{NPP}_{0,i}$ is normalized without first calculating the difference between $\text{NPP}_{0,\text{ref}}$ and $\text{NPP}_{0,i}$. This implies that the CFs of both AoP are not at all comparable and do not have to be as they represent different environmental concerns. Note that, for complete losses of NPP₀ ($a \times \text{SOC}_{\text{loss}} + b = 100$), damages only depend on the $\text{NPP}_{0,i}/\text{NPP}_{0,\text{ref}}$ ratio. As for the resource depletion indicator, regionalization was at the grid cell level, without aggregating values on broader scales. For a land use activity under study, the lower the indicator result, the less of a drop in biomass production and the less damage to the environment.

2.3 Case study on energy crops grown in Spain

The proposed method was applied to agricultural plots of food and energy crops in Spain. This was selected as a representative case study because soil loss by water erosion is one of the main causes of land degradation in the country (EEA 2005) and, important for validating the methodology, because soil erosion is unevenly distributed, with a gradient from north to south and from the Atlantic to the Mediterranean coast (MERMA 2012). Because an increase in energy crop production is forecast in Spain (EEA 2006), it is important to assess the environmental cost of this alternative, given the effect of agriculture on the environment. To illustrate the outlined method, we analyzed the environmental impacts of growing five 3-year agricultural rotations, two with energy crops, on the erosion regulation ecosystem service. The analysis includes 120 agricultural plots covering the main Spanish watersheds (ESM Fig. 2SI). The functional unit selected for the assessment was a land

occupation of 1 m^2 in 1 year (square meters per year), considering the average soil and SOC loss of a complete crop rotation system. We used the attributional framework so that any potential additional consequence of planting the rotations on other parts of the economy was not included, nor did we include past impacts on soils from historical land use. Further details on the study area, crop production systems, and data sources for the LCI are described in ESM, Sections 3 to 5.

3 Results

3.1 LCIA characterization factors

Regional characterization factors for the soil resource depletion and ecosystem quality impact pathways of the erosion regulation ecosystem service are shown in Fig. 2. GIS layers are also provided in the ESM. For soil resource depletion, the ratio is $((\text{SD}_{\text{ref}} - \text{SD}_i) / \text{SD}_{\text{ref}}) \times \text{SEF}_{\text{soil}}$. Lower soil depths and therefore higher damage factors are found at high northern latitudes and over wide areas of Asia. For ecosystem quality, Fig. 2 shows the $\text{NPP}_{0,i}/\text{NPP}_{0,\text{ref}}$ ratio. Higher biomass productivities and therefore damage factors are found at the low latitudes of the tropics. For both the soil resource depletion and ecosystem quality indicators, lower CF values indicate less sensitivity of the ecosystem to potential land use impacts.

3.2 Case study on energy crops grown in Spain

Both soil and organic carbon losses and impact factors vary as a function of location, thus leading to considerable differences in the environmental damage from soil erosion in different watersheds.

In the LCI, the crop rotation with the greatest erosion rate (Table 2) was when the field lay fallow in the last year (winter barley–winter wheat–unseeded fallow). These soil losses were around ten times higher than the crop rotation with the lowest erosion rate, the energy profitable poplar with a short forestry rotation (poplar–poplar–poplar). For annual cereal (winter barley–winter wheat–rye), legume (winter barley–winter wheat–pea), and energy crop (winter barley–winter wheat–oilseed rape) rotation systems, similar soil losses were recorded, with rates about 40 % lower than for fallow rotation. Outcomes of the LCIA phase showed the same pattern as in the LCI phase, as shown in Table 2.

According to the presented results, the watersheds in Spain with the lowest water erosion rates and environmental damage, thus making them the most appropriate for rotating poplar and oilseed rape energy crops, are the Duero (Northern Spain) and the Guadiana (Central Spain), while those with the highest water erosion rates and environmental impact are the internal watersheds of Catalonia (Northeast Spain) and the Júcar watersheds (Eastern Spain).

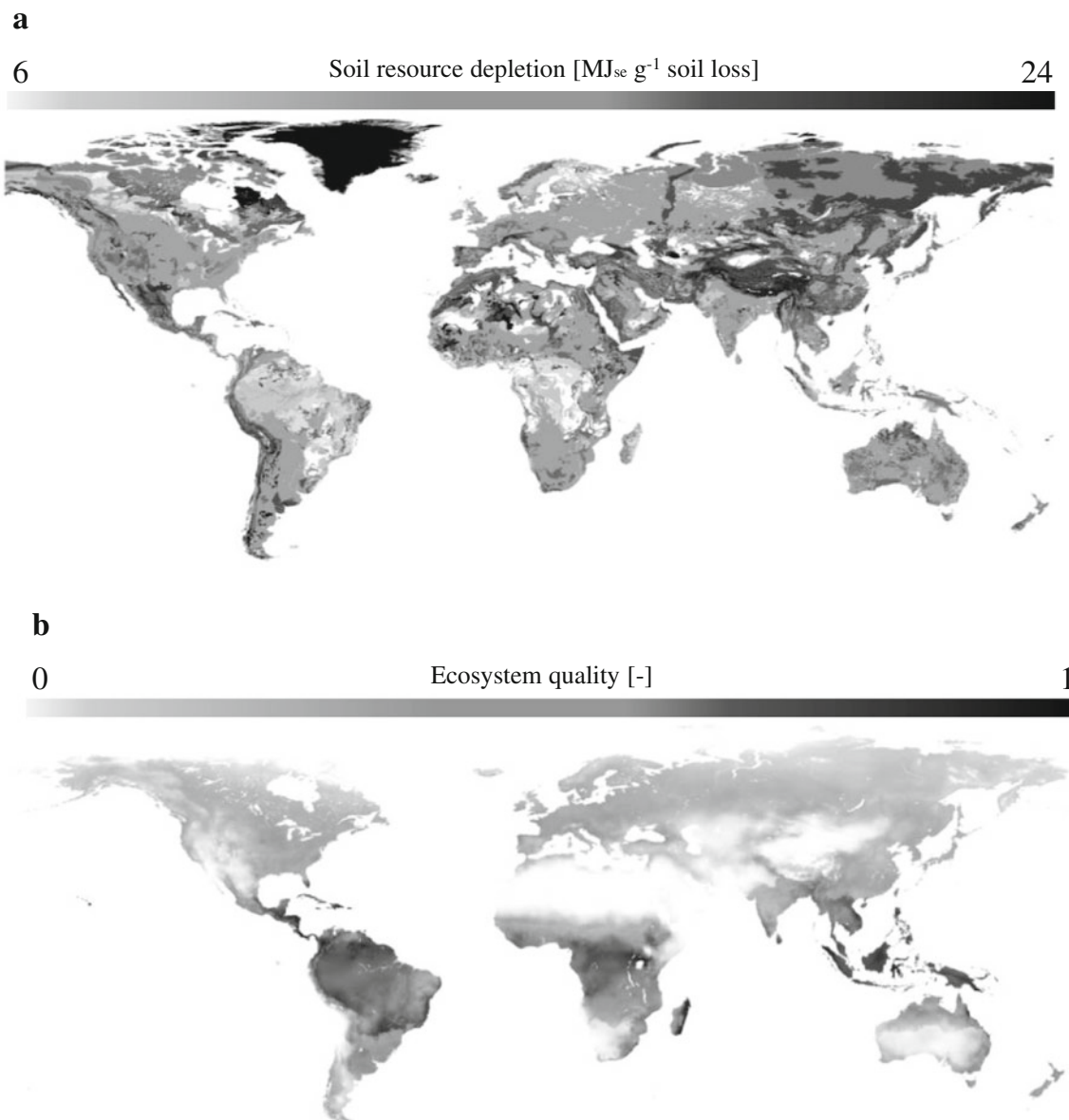


Fig. 2 Characterization factors for soil resource depletion (**a**) and ecosystem quality (**b**)

4 Discussion

4.1 Case study on energy crops grown in Spain

The results of the case study showed that the implementation of the poplar energy crop rotation system in Spain can potentially reduce erosion rates and related environmental impacts per area-time unit compared to traditional cereal and legume crop rotations in the country. In relation to the recommended locations to cultivate the analyzed crop rotations, the results indicate a trend. However, these results should be interpreted with caution. They are valid for the analyzed plots, but each specific case should be studied separately as soil erosion depends on plot-level factors. Apart from erosion, other aspects such as biodiversity impacts or

replacement of other crops, which then need to be imported, would need to be considered to assess if planting poplar in Spain is an environmentally preferable decision.

Although the area-time unit does not relate to energy crops' function of producing bio-based electricity or fuels, the framework is still meaningful when ranking crop rotations by their impact on soil replacement energy and lost productivity. Sustainable land management practices reduce impact intensity in a given area. However, intensive agricultural production can diminish soil quality and jeopardize the preservation of productive agricultural land. While impact per unit of output energy would have been a meaningful indicator to compare energy crops, this is not applicable when energy and food crops are being compared. The average results (see Table 2) are highly variable within a watershed due to the

Table 2 Life cycle inventory and life cycle impact assessment results (per square meter per year) of land occupation in ten watersheds of Spain

	Internal watersheds —Catalonia	Ebro	Duero	Júcar	Tajo	Guadiana	Segura	Guadalquivir	Mediterranean— Andalusia	Atlantic— Andalusia
LCI—soil erosion (10^3g)										
B-W-R	1.58	0.43	0.27	1.13	0.56	0.31	0.39	0.68	0.69	0.59
B-W-P	1.63	0.45	0.29	1.16	0.58	0.31	0.39	0.68	0.68	0.58
B-W-F	2.44	0.70	0.46	1.78	0.90	0.50	0.60	1.07	1.01	0.85
B-W-OR ^a	1.51	0.41	0.24	1.03	0.54	0.28	0.38	0.63	0.63	0.56
PP ^a -PP ^a -PP ^a	0.28	0.09	0.05	0.20	0.10	0.06	0.07	0.11	0.09	0.08
Soil resource depletion ($\text{MJ}_{\text{sc}}\text{m}^{-2}\text{year}^{-1}$)										
B-W-R	2.5E+04	6.4E+03	4.1E+03	1.9E+04	9.0E+03	5.0E+03	5.8E+03	9.9E+03	9.7E+03	9.9E+03
B-W-P	2.6E+04	6.6E+03	4.4E+03	2.0E+04	9.3E+03	5.2E+03	5.8E+03	9.8E+03	9.7E+03	9.9E+03
B-W-F	3.9E+04	1.0E+04	7.0E+03	3.0E+04	1.4E+04	8.2E+03	9.2E+03	1.6E+04	1.5E+04	1.4E+04
B-W-OR ^a	2.4E+04	6.0E+03	3.6E+03	1.8E+04	8.7E+03	4.6E+03	5.6E+03	9.2E+03	9.2E+03	9.4E+03
PP ^a -PP ^a -PP ^a	4.5E+03	1.3E+03	7.5E+02	3.4E+03	1.6E+03	9.0E+02	9.7E+02	1.6E+03	1.4E+03	1.3E+03
Ecosystem quality ($\text{NPPD m}^{-2}\text{year}^{-1}$, decimal % of NPP_0 lost $\text{m}^{-2}\text{year}^{-1}$)										
B-W-R	0.12	0.03	0.02	0.07	0.04	0.03	0.02	0.05	0.04	0.05
B-W-P	0.12	0.04	0.02	0.07	0.04	0.03	0.02	0.05	0.04	0.05
B-W-F	0.18	0.05	0.03	0.11	0.06	0.04	0.03	0.07	0.06	0.07
B-W-OR ^a	0.11	0.03	0.02	0.07	0.04	0.02	0.02	0.05	0.04	0.05
PP ^a -PP ^a -PP ^a	0.03	0.02	0.01	0.02	0.01	0.01	0.01	0.02	0.02	0.02

B-W-R winter barley–winter wheat–rye, *B-W-P* winter barley–winter wheat–pea, *B-W-F* winter barley–winter wheat–unseeded fallow, *B-W-OR* winter barley–winter wheat–oilseed rape, *PP-PP-PP* poplar–poplar–poplar

^a Crops for energy use

disparity of soils, climates, and ecosystem biomass productivities (see ESM Section 6SI). There is a need for statistical analyses combining watersheds and geospatial features to show to what extent the average results can be extrapolated across the watershed. Aggregation at the watershed level was not specific enough to reflect a common trend, though it is a useful reference area to compare to and/or combine with the results of a water use impact assessment. Further highlights of the results are discussed in ESM Section 7.

4.2 Soil erosion impact assessment model

Analysis of the method followed the general evaluation criteria and specific sub-criteria relevant for land use impacts in the ILCD Handbook (JRC 2010). The aim of the analysis was to qualitatively address model uncertainty, facilitate comparison with other soil erosion impact assessment models, and identify the strengths and weaknesses of our method.

4.2.1 Completeness of scope

Two relevant impact pathways leading to AoP natural resources and natural environment were addressed at the endpoint level. We expressed ecosystem damages using a growth-based value (NPPD), while the majority of endpoint methods express ecosystem impacts as PDF (Goedkoop and Spriensma 2001;

Goedkoop et al. 2009; Jolliet et al. 2003). A significant correlation has been found (Pfister et al. 2009) between vascular plant species biodiversity and NPP of the actual vegetation, which led the authors to select NPP as a proxy for ecosystem quality. The same proxy could be applied to transform NPPD to PDF, taking into consideration that we used potential instead of actual vegetation. Globally, the NPP_0 -to- NPP_{act} ratio ranges from 0.9 to 1.2 on 81 % of the terrestrial area.

Similarly, for the indicator of soil resource depletion, we used energy units, as in the work by Rugani et al. (2011), whereas most endpoint resource demand indicators are expressed in energy units (in megajoule), as in the EI99 (Goedkoop and Spriensma 2001) and IMPACT2002+ (Jolliet et al. 2003) methods. In reality, any energy measure for reconstructing a renewable resource such as soil lacks meaning. To allow for comparisons with other methods and impact categories, energy values can be transformed into emergies using the resource-specific solar equivalent factor. When this is done, the results can be compared to or aggregated with surplus emergy demands of energy carriers (e.g., crude oil, $0.091 \text{ MJ}_{\text{sc}}\text{g}^{-1}$) as well as non-energetic resources (e.g., water supply, $0.203 \text{ MJ}_{\text{sc}}\text{g}^{-1}$; Zhang et al. 2010). We assumed an SEF_{soil} of $23.9 \text{ MJ}_{\text{sc}}\text{g}^{-1}$ soil loss (Odum 1996) as this is the only available estimate. This global value does not distinguish between different soil types, land uses, and world regions, so it should be further refined.

One main soil erosion-related impact pathway for human health was identified (impact pathway 3 in Fig. 1). This is discussed, although not modeled, in ESM Section 8.

The characterization model and factors are globally applicable and spatially defined, taking into account ecosystem biomass productivity and soil characteristics at the grid level. How to aggregate these grid cell-specific factors on a wider and still accurate scale (e.g., ecoregions, land cover, watersheds) is a complex unresolved issue due to the huge variability of soil types even at the landscape scale. This variability of soil types makes it difficult to identify a standardized approach to address spatial differentiation.

4.2.2 Environmental relevance

The environmental impacts of soil erosion are manifold and affect both human life and ecosystems; hence, soil erosion control is an important ecosystem service that should be considered in the evaluation of the environmental sustainability of land use activities. The method we developed focuses on the assessment of land occupation impacts (i.e., the use of a land area for a specific human purpose) on soil loss for any type of human activity, whenever the soil erosion rate of the assessed land use activity (e.g., industrial and mining) is available or estimated by the LCA practitioner. However, the method discounts impacts due to land transformation (i.e., change of a land area to make it suitable for a specific use). For occupation, it is debatable what period of occupation should be considered in the agricultural LCA inventory (i.e., duration of the crop, duration of the crop plus the fallow period, and crop rotation). For both transformation and occupation, the choice of the reference situation to measure the magnitude of the change and the time needed after occupation to recover this reference situation are two fundamental issues to be agreed upon to properly assess impacts from both land use interventions. So far, the recommended reference situation in attributional LCA is the so-called potential natural vegetation after land occupation (Milà i Canals et al. 2007a), which differs from the natural situation because nature rarely returns to its original state after being disturbed.

Of the overall cause–effect chain for land use (see Fig. 1), we focused on the soil erosion impact mechanism by quantifying changes in topsoil preservation and changes in the NPP₀ due to altered soil function. For a complete evaluation of land use impacts, the soil erosion assessment must be complemented with indicators that measure effects on climate change, biodiversity loss, water and nutrient regulation, and unique landscapes.

4.2.3 Scientific robustness and certainty

The two indicators reflect the cause–effect chain from interventions to the final environmental damages. For soil

resource depletion, the impact can be expressed at the mid-point level if soil losses of the LCI are weighted by the available reference-corrected soil depth without further transformation to emergy units. The cause–effect chain for ecosystem quality directly links soil erosion to the NPP₀ depletion indicator, which can be further modeled to PDFs.

The geographical differentiation of the model has good potential for being improved and further developed when more detailed global maps of soil properties and biomass productivity of ecosystems become available. The current large-scale maps do not fully resolve the real diversity of soil and ecosystems.

The indicator for ecosystem quality can be partially verified against monitoring data by measuring the loss of NPP at different degrees of soil erosion. Note that we used potential NPP, so the model can be tested in areas with potential or near potential vegetation. The indicator for soil resource depletion cannot be verified (MJ_{se}) as emergy is an abstract concept.

Model uncertainties were qualitatively evaluated using the set of criteria listed in the ILCD Handbook (JRC 2010). Most statistical and decision rule uncertainties still have to be estimated. Considerable statistical uncertainties occur in the ecosystem quality impact assessment model. Our approach of linking soil erosion to NPP loss is based on limited and highly uncertain studies about the effect of soil erosion on an ecosystem's biomass production. This adds substantial uncertainty to our model. Moreover, we allocated quantitative ranges of soil loss to qualitative classes of soil loss (ESM Table 2SI) and then calculated a linear regression with the average of each range. Choosing other quantitative ranges or a value different from the average of each range would have given different NPPD equations. Another source of uncertainty arises from the type of relationship established between SOC losses and NPP₀ losses. We assumed that both variables are linearly related. The linear model was used instead of nonlinear models because there is insufficient knowledge of the type of relation between the two variables and the possible interference of other variables (e.g., climate change).

Further statistical uncertainty that affects input data in the LCI and the characterization factors is the uncertainty arising from the resolution of soil data. Also, for the soil resource depletion indicator, the use of a site-dependent solar energy factor would reduce uncertainty. Uncertainty also comes from the type of spatial aggregation of the LCI and characterization factor results. These are key future research directions to improve the reliability of the model.

4.2.4 Documentation, transparency, and reproducibility

The documentation used for the model is published and readily accessible. The maps used to derive the characterization factors are available online on valid and reliable sources of

data, namely, the FAO's site (FAO/UNESCO 2007) and in published, peer-reviewed, scientific articles (Haberl et al. 2007). This availability of input data allows third parties to further develop and improve the impact factors and the model.

4.2.5 Applicability

At the present state of development, two obstacles hinder the applicability of the method across the overall life cycle of a product. At the inventory stage, unit processes of existing LCI datasets currently lack soil loss figures. This makes it difficult to include the background system in the soil erosion assessment. At the impact assessment stage, we regionalized CFs on a gridded spatial scale, while CFs should be aggregated to cope with data gaps of location of processes. To overcome these problems, processes of the LCI datasets should include information on soil erosion, and characterization factors, currently disaggregated at the grid cell scale, should be aggregated on larger scales. A site-generic CF, which is useful for processes of unknown location, may be derived by aggregating all the gridded-based CFs. The freshwater ecoregion regionalization approach, as suggested in Koellner et al. (2012), might be an option of aggregation to be studied in future work.

The CFs are applicable by general LCA practitioners. Incorporation in current LCA software will require adaptation of the software to accommodate spatially explicit data in GIS format. Estimates of the soil erosion data needed for LCI and measured with USLE require some knowledge of soil sciences and experience in the application of the equation.

4.2.6 Stakeholder acceptance

For the ecosystem quality indicator, the results are easily interpretable and understandable by non-LCA experts. In contrast, a possible barrier hindering acceptance of the soil resource depletion indicator is the use of surplus emergy units, which may be deemed too complex and meaningless for most people. The absence of soil and climate data availability can also hamper the applicability and acceptance of the two indicators.

5 Conclusions

We developed a globally applicable, spatially differentiated LCIA method to account for land occupation impacts in LCA, focusing on the aspect of soil erosion. The LCI data required and data sources and models that can be used to obtain the inventory flows were also identified. Spatially explicit damage factors on a grid cell-level resolution (approximately $10 \times 10 \text{ km}^2$) for the entire world were provided for soil resource depletion and ecosystem quality endpoints.

The LCA model was successfully applied to agricultural plots in Spain to compare soil erosion-related environmental impacts that may result from substituting traditional food for energy crop rotations. Results from the case study show that the lowest erosion rates and environmental damage occur when rotating poplar short rotation coppice in Northern and Central watersheds in Spain.

Further research should focus on applying the method across the overall life cycle of a product and to identify a feasible and relevant spatial scale at which to aggregate characterization factors to cope with data gaps on the location of processes. Other areas for future research deal with improving the reliability of the model by quantifying and, insofar as it is possible, reducing statistical uncertainty arising from the key sources discussed in the article.

Acknowledgments This work was carried out within the framework of the national and strategic On Cultivos Project, funded by the Spanish Ministry of Science and Innovation and the European Regional Development Fund, and the LC-IMPACT project—Improved Life Cycle Impact Assessment Methods (LCIA) for Better Sustainability Assessment of Technologies (grant agreement no. 243827), funded by the European Commission under the 7th Framework Programme on the Environment, ENV.2009.3.3.2.1. We would like to thank the staff at IRTA-Experimental Station Mas Badia Foundation (Spain) and Dr. Asunción Usón for their help with the case study.

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